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**EFFECTS OF ALUM TREATMENT
ON PHOSPHORUS DYNAMICS
AND PHYTOPLANKTON RESPONSE
IN EAU GALLE RESERVOIR, WISCONSIN**

by

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<p>The hypolimnion of Eau Galle Reservoir, Wisconsin, was treated with aluminum sulfate (alum) in 1986 to reduce excessive phytoplankton production associated with high phosphorus loading from anoxic, profundal sediments. Prior to treatment, internal total phosphorus (TP) loading was 3 to 6 times greater than external TP loading during summer stratification. Periodic increases in epilimnetic TP mass and chlorophyll α concentrations closely corresponded with elevated internal TP. For 1 year following treatment, internal TP loading and concentrations of soluble reactive phosphorus in the hypolimnion were substantially reduced.</p> <p>Abnormally high external TP loading during the stratified period of 1986 sustained high TP mass and chlorophyll in the epilimnion. During the summer of 1987, effects of alum treatment on internal TP loading were essentially negated, as epilimnetic TP mass and</p> <p style="text-align: right;">(Continued)</p>					
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chlorophyll remained unchanged from pretreatment years. Sedimentation of external TP inputs, with subsequent alum burial, is believed to have been primarily responsible for reducing the effectiveness of alum in controlling internal TP loading in this reservoir.

Following alum treatment in 1986, phytoplankton biomass was less than half that occurring during the previous year; however, it was not significantly different from other years. Subsequent to several years of codominance with blue-green algae, the dinoflagellate *Ceratium hirundinella* assumed singular dominance in 1985. With the exception of this change in relative dominance (maintained through 1988), the phytoplankton community of Eau Galle Reservoir has been remarkably stable, despite effects of a winter drawdown in 1984, alum treatment in 1986, and climatic (wet versus dry) extremes over the 8-year study period. Without a decrease in phosphorus availability, *Ceratium* and blue-green algae are likely to persist as codominants during the summer in this system for many years.

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PREFACE

The studies reported herein were sponsored by the Headquarters, US Army Corps of Engineers (HQUSACE), Directorate of Civil Works, through the US Army Corps of Engineers Environmental and Water Quality Operational Studies (EWQOS) Program, the Algae Control Demonstration (ACD) Program, the Water Operations Technical Support (WOTS) Program, and the US Army Engineer District (USAED), St. Paul, St. Paul, MN. The EWQOS and ACD Programs were managed by the US Army Engineer Waterways Experiment Station (WES), under the direction of Dr. Jerome L. Mahloch, Manager. The WOTS Program is managed by the WES under the Environmental Resources Research and Assistance Programs, Mr. J. Lewis Decell, Manager. Technical Monitors for the study were Dr. John Bushman, Mr. James Gottesman, and Mr. David Buelow, HQUSACE.

Principal investigator for this study was Dr. John W. Barko, Environmental Laboratory (EL), WES. Experimental design, data analysis and interpretation, and preparation of the report were accomplished by Drs. Barko, William F. James, and William D. Taylor and Ms. Dwilette G. McFarland. Technical reviews were provided by Drs. Robert Gaugush, Douglas Gunnison, Thomas L. Hart, and Nancy J. McCreary of the EL. Additional reviews were provided by Messrs. Daniel Wilcox and Dennis Holme of the USAED, St. Paul.

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This investigation was performed under the general supervision of Dr. John Harrison, Chief, EL, and Mr. Donald L. Robey, Chief, Ecosystem Research and Simulation Division, and under the direct supervision of Dr. Thomas L. Hart, Chief, Aquatic Processes and Effects Group.

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CONTENTS

	<u>Page</u>
PREFACE.....	1
PART I: INTRODUCTION.....	4
Background.....	4
Site Description.....	5
Objectives.....	6
PART II: METHODS.....	7
Alum Treatment.....	7
Analytical Protocol.....	7
Phosphorus Loading Determinations.....	8
Phytoplankton Determinations.....	9
PART III: RESULTS.....	11
Phosphorus Dynamics.....	11
Phytoplankton Response.....	13
PART IV: DISCUSSION.....	16
PART V: FUTURE STUDIES.....	22
REFERENCES.....	23
TABLES 1-4	
FIGURES 1-11	
APPENDIX A: PHYTOPLANKTON SPECIES LIST, EAU GALLE RESERVOIR.....	A1

EFFECTS OF ALUM TREATMENT ON PHOSPHORUS DYNAMICS AND PHYTOPLANKTON
RESPONSE IN EAU GALLE RESERVOIR, WISCONSIN

PART I: INTRODUCTION

Background

1. Eutrophication is a process frequently stimulated by the transport of nutrients from watershed to lake. While this is a natural process in all lakes and reservoirs, the rate at which it occurs is often accelerated by human activities in the watershed. Symptomatic of the advancement of eutrophication are reduced water clarity, poor dissolved oxygen conditions in bottom waters, excessive algal production, and impaired recreational value.

2. The obvious best means of ameliorating these undesirable conditions is the reduction of external material inputs. Many lake restoration techniques have been directed toward the control of external phosphorus (P) loading to reduce phytoplankton production. Successful lake restoration has often been achieved by diverting stormwater, sewage, and septic tank inputs (e.g., Edmondson 1970, Cooke et al. 1978). However, internal P loading can significantly delay lake recovery following implementation of these restoration efforts (Welch 1977, Larson et al. 1979, Ryding 1981).

3. Concentrations of P usually increase with increasing depth in the hypolimnion, a pattern closely related to changes in the area-to-volume ratio with depth and the progression of anoxia (Stauffer 1985). An increasing body of evidence indicates that anoxic, profundal sediments are an important source of internal P loading and contribute significantly to patterns of increase in P mass in the hypolimnetic water column (Mortimer 1971; Cooke et al. 1977; Theis and McCabe 1978; Larson, Schultz, and Malueg 1981; Nurnberg 1984, 1987; Riley and Prepas 1984; Nurnberg et al. 1986).

4. Phosphorus released from anoxic sediments may be transported to the epilimnion via thermally induced turbulent eddy diffusion (Jassby and Powell 1975; Imboden and Emerson 1978; Robards and Ward 1978; Stefan and Hanson 1981; Wodka et al. 1983) and wind-induced metalimnetic migration (Stauffer and Lee 1973; Larson, Schultz, and Malueg 1981; Kortmann et al. 1982; Stauffer and Armstrong 1984; Stauffer 1985, 1987; Effler et al. 1986; James, Kennedy, and

Gaugush, in press). These processes stimulate phytoplankton productivity, and thereby reduce recreational value. Under such circumstances, alternative means for improving or maintaining water quality must be sought. Many of these alternative solutions involve attempts to control the availability of P within the water body. With decreased availability of P, excessive phytoplankton growth can usually be curtailed.

5. Ferric aluminum sulfate and aluminum sulfate (alum) treatments have been widely used in lakes to reduce internal P loading from the sediment, and thereby reduce phytoplankton biomass (Peterson et al. 1973; Cooke and Kennedy 1978; Cooke et al. 1978, 1982; Soltero et al. 1981; Garrison and Knauer 1983; Foy 1985). The success of alum treatment usually depends on concurrent reduction of external P loading in addition to long-term control of sediment P release (Kennedy and Cooke 1982). While most natural lakes receive the majority of their P income from internal sources during the summer, reservoirs often receive substantial external P inputs from inflowing tributaries. Thus, in reservoirs, the success of alum treatment may largely depend on the seasonality of external P inputs.

Site Description

6. Eau Galle Reservoir (Figure 1) is a small (0.6-km^2) impoundment located in west-central Wisconsin. It is operated by the US Army Engineer District, St. Paul. The reservoir is dimictic and approximately circular in shape. It has a mean depth of 3.2 m, a maximum depth of 9 m, and receives the majority of its water (80 percent) and nutrient inputs from the Eau Galle River. Minor inflowing tributaries include French, Lousy, and Lohn Creeks.

7. External P loading cannot be controlled effectively in this reservoir because of its large size (166 km^2) and the agricultural nature of the watershed. External loadings occur primarily in the spring and fall, during which periods the entire volume of the reservoir can be replaced within only a few days. Loadings are usually minimal during the summer months, although freshets do occur with low frequency (Kennedy 1987). Internal P loading from anoxic sediments and the accumulation of hypolimnetic P is substantial during the summer (James, Kennedy and Gaugush, in press).

8. The outlet structure has both surface and hypolimnetic discharge ports. Surface discharge is uncontrolled and occurs at pool elevations above

286.5 m, relative to mean sea level. Discharge during the summer stratified period occurs primarily through a controlled vertical slide gate located 7.2 m below the surface discharge elevation. During the winter of 1984, the reservoir was drawn down to nearly 1 m below normal pool elevation over about a 4-month period for maintenance purposes. This event exposed variably up to 53 percent of the reservoir littoral zone to freezing and desiccation, and accounted for about a 47-percent loss of pool volume.

9. Withdrawal of cooler water during the summer stratified period gradually warms the hypolimnion, promoting a decrease in the stability of the water column (Gaugush 1984) with associated increases in phytoplankton production, primarily blue-green and dinoflagellate populations (Barko et al. 1984). During periods of declining stability in the summer, migrations of the metalimnion occur in Eau Galle Reservoir as a result of wind mixing (James, Kennedy, and Gaugush, in press). Mixing events have an important influence on P availability to phytoplankton in the epilimnion of this system.

Objectives

10. The objective of this investigation was to limit phytoplankton population densities by controlling (via alum treatment) internal P loading from profundal sediments. To our knowledge, direct control of sediment P release with alum has heretofore not been attempted in flood-control reservoirs. The studies reported herein examine the effects of a hypolimnetic alum treatment on sediment P dynamics and phytoplankton response. Results are reported for the pretreatment (1981-1985) and posttreatment (1986-1988) periods. This report provides extensive information on P cycling and contains a detailed account of phytoplankton community composition in Eau Galle Reservoir over the 8-year study period.

PART II: METHODS

Alum Treatment

11. Alum (11.3 g Al/m^2 ; 4.5 mg Al/l) was administered to the reservoir on 29 May 1986. Details of dosage and application procedures are provided in Kennedy et al. (1987). Based on previous measurements of internal total phosphorus (TP) loading, the dose applied was theoretically sufficient to control sediment P release for 5 years. The treatment area was confined to depths of 3 m and greater, since sediments at these depths experience anoxia during summer stratification. Here, only anoxic sediments were assumed to contribute to the internal P load.

Analytical Protocol

12. Water samples for chemical analyses were collected at 1- or 2-week intervals at 1-m increments throughout the summer months in 1981, 1982, 1986, 1987, and 1988 at a deep (9-m), centrally located station. The samples were collected with either an electric pump and hose or a Van Dorn water sampler. Additional samples were collected at 25-cm intervals from the reservoir surface to a 3-m depth in 1987, and from the 7-m depth to the reservoir bottom in 1982, 1986, 1987, and 1988. A small-volume Masterflex peristaltic pump and tubing (6-mm inside diameter) were used to collect these samples near the reservoir surface, and a pneumatically driven close-interval sampler (Blakar 1979) was used to collect samples between 7 m and the sediment surface at approximately 9 m.

13. River water samples for chemical analyses were collected weekly (or more frequently) at fixed stations located on the Eau Galle River, Lohn Creek, Lousy Creek, French Creek, and the reservoir tailwater. In 1987 and 1988, samples from the Eau Galle River and the tailwater were collected on nearly every working day throughout the summer. In addition, samples were collected on the Eau Galle River during 1987 at hourly intervals using an ISCO automated water sampler during all storm periods.

14. Samples for total soluble phosphorus (TSP) and soluble reactive phosphorus (SRP) were filtered through a $0.45\text{-}\mu$ membrane filter before analysis. Samples collected under anoxic conditions were filtered in situ with

syringe filters (Nalge). Total phosphorus and TSP were determined colorimetrically on a Technicon Auto-Analyzer II following persulfate oxidation (APHA 1985). Soluble reactive phosphorus was determined using the ascorbic acid method on either a Technicon Auto-Analyzer II or a spectrophotometer (APHA 1985). The detection limit for P analyses was 0.005 mg/l. Samples for chlorophyll *a* determination were filtered onto glass fiber filters and extracted in 90-percent alkaline acetone (APHA 1985). Chlorophyll *a*, corrected for phaeopigments, was calculated according to Lorenzen (1967). Statistical analyses were performed using the Statistical Analysis System (SAS 1985).

Phosphorus Loading Determinations

15. The Eau Galle River and tailwater flows were gauged continuously except during 1981. Flows from French, Lousy, and Lohn Creeks were determined from continuous-recording gauges or stage height recorders during all years except 1981 and 1988. Reservoir level (for reservoir volume determination) was measured daily with a recording gauge. Daily precipitation (in centimeters) was obtained from the nearby Eau Galle Recreation Area, Resource Manager's Office.

16. A hydrological mass balance was determined for the summer stratified period for all years except 1981. The change in storage of water mass was assessed according to the following mass balance statement:

$$\begin{aligned} \text{Change in Storage} = & (\text{Gauged Inflows} + \text{Precipitation}) \\ & - (\text{Gauged Outflow}) \pm (\text{Ungauged Inflows and Outflows}) \end{aligned} \quad (1)$$

Ungauged inflows and outflows collectively comprised ground water, overland runoff, and evaporation. These sources were not directly measured, but were determined from the above equation.

17. Daily external TP loadings, determined for 1987 and 1988, were calculated by multiplying the inflow rate by the measured TP concentration. Daily external TP loadings for 1982 and 1986 (sampled only weekly), and for unsampled days during 1987 and 1988, were estimated using regression analyses (Method 4, Walker 1985) and relationships between measured daily TP

concentration and flow determined for 1987. Internal TP loading was determined according to the following mass balance equation:

$$\begin{aligned} \text{Change in TP Mass} = & (\text{External TP Load} - \text{TP Outflow}) \\ & + \text{Internal TP Load} \end{aligned} \quad (2)$$

Internal TP loadings were calculated for the years 1981, 1982, 1986, 1987, and 1988. The reservoir TP mass was determined as the sum of volume-weighted TP mass of the water column. The TP mass was also calculated separately for the 0- to 3-m depths (i.e., epilimnion) and for the 3- to 9-m depths (i.e., metalimnion and hypolimnion).

Phytoplankton Determinations

18. Phytoplankton sampling has been uninterrupted since 1981. During most years, epilimnetic water samples for phytoplankton identification and enumeration were collected at 2-week intervals between the onset of anoxia (May-June) and autumnal overturn (August-September). In 1986 these samples were collected weekly. During other seasons, sampling frequency varied between 2 and 6 weeks. In 1981, phytoplankton samples were collected from six stations. Since it was determined that the station located over the central depression was representative of the pelagic zone (Barko et al. 1984), only phytoplankton data from that station are reported here.

19. Integrated water samples were taken using a tubular collecting device, constructed from a 7.6-cm-diam plastic pipe and equipped with a one-way check valve. The check valve allowed filling of the pipe during vertical immersion and sample retention upon withdrawal from the water. Samples were taken to a depth of 3 m. Subsamples (125 ml) were preserved with acid-Lugol's solution (Vollenweider 1969) and stored in amber glass bottles at room temperature until processed. Taxonomic identification and enumeration of phytoplankton were accomplished using phase contrast inverted microscopes (Wild M40 and Leitz Diavert) in general accordance with Lund, Kipling, and LeCren (1958). Specific enumeration procedures are described in Barko et al. (1984).

20. Permanent slides of diatoms were prepared using Hyrax medium following digestion in boiling concentrated HNO_3 . These slides were examined

using a Bausch and Lomb phase contrast microscope to estimate the relative abundance of different diatom taxa enumerated by inverted microscopy. The contribution of dead cells to the diatom counts was minor.

21. Cellular volume was computed from average cell dimensions determined for 25 randomly selected cells of the more common species from each sample (Munawar and Munawar 1976). Volume calculations assumed general conformity to the geometry most closely resembling that of specific algal cells (Vollenweider 1969). Cell volume was converted to fresh weight biomass assuming that the specific gravity of phytoplankton approximates 1.0 (Nauwerck 1963). Phytoplankton biomass is reported as grams per cubic meter fresh weight.

22. Species diversity (H'), after Shannon and Weaver (1963), was calculated based on fresh weight as follows:

$$H' = - \sum_{i=1}^S P_i * \ln P_i \quad (3)$$

where

P_i = proportion of the i^{th} taxon in the sample, calculated from n_i/N

n_i = biomass of individuals per cubic meter of the i^{th} taxon

N = total biomass per cubic meter

Evenness (J), a measure of how closely H' approaches the maximum possible diversity, was calculated (after Pielou 1966) as

$$\frac{H'}{\ln S} \quad (4)$$

where

S = number of taxa present in sample

$\ln S$ = maximum diversity

PART III: RESULTS

Phosphorus Dynamics

23. Large between-year differences were observed in inflow, outflow, precipitation, hydraulic residence time, and flushing rate during the stratified period (Figure 2). The Eau Galle River was the major tributary input, accounting for nearly 80 percent of the total gauged inflow during all years. Mean daily gauged inflow (including the Eau Galle River and minor tributaries) and outflow were about three times higher in 1986 (the year of alum treatment) than in other years, due primarily to the large amount and frequent periods of precipitation that occurred during that year. Ungauged inflows and outflows were greater in 1986 than in other years. Also during that year, the hydraulic residence time was greatly reduced, and the flushing rate was elevated relative to all other years. Major freshets caused peaks in external TP loading from the Eau Galle River, primarily in 1986 but also during late summer of 1987 (Figure 3). Overall, external TP loading was relatively low during most of the summer of 1987. The lack of summer storm events in 1982 and 1988 (drought years) resulted in extremely low external TP loadings.

24. Eau Galle Reservoir commonly exhibited elevated concentrations of SRP in the hypolimnion before alum treatment, as in 1982 (Figure 4). Shortly after the occurrence of anoxic conditions, concentrations of SRP increased above the sediment surface, indicating P release from anoxic sediment. Steep vertical gradients in SRP were observed, and the concentration exceeded 1.00 mg P/l within the bottom meter from June through September of 1982. Hypolimnetic withdrawal appeared to result in lower SRP accumulation above the 7-m depth. Declines in SRP at intermediate depths in June and early August of 1982 coincided with decreases in the volume of anoxic water due to mixing.

25. Alum treatment resulted in low SRP throughout the water column from June until late July 1986 (Figure 4). Steep vertical gradients in SRP again developed below the 6-m depth during late August 1986. However, absolute SRP concentrations in the hypolimnion were much lower in 1986 than in 1982. Slight increases in SRP at intermediate depths (4 to 6 m) in mid-August 1986 coincided with high external TP loading (Figure 3). In 1987 and 1988, the patterns of SRP increase were very similar to those observed in 1982. Periods

of major external TP loading caused increases in SRP throughout the water column in September 1986, August 1987, and September 1988.

26. During the stratified periods of 1981 and 1982, internal TP loading frequently exceeded $10 \text{ mg/m}^2/\text{day}$ (Figure 5) and was 3 to 6 times greater than external TP loading (Table 1). During these years, internal TP loading was greatest when the zone of anoxia was large, and least during periods of mixing. The passage of cold fronts with sustained high winds caused mixing, which reduced the zone of anoxia (Figure 4). Total phosphorus discharge via withdrawal accounted for a large portion (20 to >100 percent) of the total TP load (i.e., internal plus external TP loading) before treatment (Figure 5).

27. Alum treatment completely negated internal TP loading from June to mid-July 1986 (Figure 5). During that period, negative internal TP loading values reflect losses through sedimentation and/or discharge. Unfortunately, separate sources (i.e., internal versus external) of TP discharged from the reservoir could not be determined. Positive internal TP loading resumed in late July and August 1986. In 1987 and 1988, internal TP loading was similar to pretreatment years (1981 and 1982) and accounted for 58 to 85 percent of the total TP load to the reservoir during summer stratification (Table 1).

28. Before alum treatment, increases in epilimnetic TP mass (Figure 6) corresponded with increased internal TP loading (Figure 5), as during mid-June and mid-July 1981 and during late July and late August 1982. Similarly, during the 1988 drought (after alum treatment), epilimnetic TP mass exhibited increases in June and July, during periods of elevated internal TP loading. In contrast, during late July and late August 1986, the year of alum treatment, external TP loading appears to have been the major contributor to TP mass increases in the epilimnion. In 1987, both internal and external TP sources appear to have contributed to increases in TP mass in the epilimnion. The TP mass in the epilimnion increased in early July of that year, corresponding to increased internal TP loading (Figure 5). However, during late July and early August 1987, increases in epilimnetic TP mass corresponded closely with periods of elevated external TP loading.

29. Before alum treatment, TP mass in the reservoir bottom waters (i.e., metalimnion and hypolimnion) generally increased during or shortly before corresponding increases in TP mass in the epilimnion (Figure 6), suggesting vertical transport of TP. The TP mass in the bottom waters was greater than in the epilimnion before alum treatment (Figure 6, 1981 and

1982). In 1986, however, TP mass in the bottom waters decreased substantially in relation to epilimnetic TP mass due to the effects of alum treatment. The epilimnion and the reservoir bottom waters exhibited similar TP masses during the early through mid-summer months of 1987 and 1988. During these same years, TP mass in the bottom waters exceeded that of the epilimnion from late July through August.

30. Peaks in epilimnetic (0 to 3 m) chlorophyll α concentration (Figure 7) corresponded both with peaks in epilimnetic TP mass (Figure 6) and with periods of elevated internal or external TP loading (Figures 3 and 5). Chlorophyll α increases in July of 1981, 1982, 1987, and 1988 were associated with periods of elevated internal TP loading. Increases in chlorophyll α during August of 1986 and 1987 coincided with periods of elevated external TP loading. In general, alum treatment had no significant effect on mean chlorophyll α concentration and mean TP mass of the epilimnion (Table 2).

Phytoplankton Response

31. The seasonal distribution of phytoplankton biomass is presented in Figure 8 for the 8-year study period. Generally, a spring diatom (Bacillariophyceae) bloom was followed by a clear-water phase prior to the development of a large summer population dominated by dinoflagellates (Pyrrhophyta) and blue-green algae (Cyanophyta). Spring biomass maxima varied greatly in magnitude, duration, and timing. The largest occurred in April 1981 (66 g/m^3 fresh weight) and the smallest in March 1987 (6 g/m^3). Frequently, a second diatom bloom developed during the autumn. Cryptophytes (Cryptophyta) were occasionally as important as diatoms in the spring (1984) and autumn populations (1983, 1984, 1986, and 1987).

32. Major summer biomass peaks occurred each year between about the third week in June and the end of September (Figure 8). In five of the studied years, biomass declined abruptly in October. However, in 1981 and 1982, substantial phytoplankton biomass remained beyond October, and in 1988, biomass in the form of a blue-green algae bloom (*Aphanizomenon flos-aquae*) was increasing when sampling ended in October. Maximum summer biomass ranged from 26 g/m^3 in late September 1984 to 89 g/m^3 in July and August 1985. Maximum summer biomass occurred as early as the third week of June 1988 and as late as the third week of September 1984. Average summer biomass ranged between

10 and 20 g/m³ during all years except 1985, during which biomass was much greater at 51 g/m³ (Figure 9). The dinoflagellates produced more biomass than the blue-green algae in all study years except 1981 and 1984 (Figure 8). Following reservoir drawdown during the winter of 1984, mean summer blue-green algae biomass was reduced by about one half in 1985. This reduction coincided with an increase in dinoflagellate biomass during the same year.

33. Between-year differences in phytoplankton succession during the summer growing season are summarized in Figure 10. Dinoflagellates and blue-green algae have been the perennial summer dominant groups. Only did 1981 did blue-green algae establish dominance earlier in the season than did dinoflagellates. Transitions in community structure during the summer have demonstrated no consistently recurring patterns. However, between 1985 and 1987, dinoflagellates maintained dominance throughout the summer. In this reservoir, for all practical purposes, *Ceratium* biomass is synonymous with dinoflagellate biomass.

34. A listing of the 136 phytoplankton taxa identified in the course of this 8-year study is provided as Appendix A. This is a record of the more abundant and commonly encountered species rather than an exhaustive list. Relatively few phytoplankton species accounted for most of the biomass during the year (Table 3). Throughout the study, 21 phytoplankton taxa attained biomass levels greater than 1 g/m³ on at least one occasion. Of these, only the first eight species (Table 3) made consistent contributions during each study year.

35. *Stephanodiscus hantzschii* was by far the most important spring species, always accompanied by *S. invisitatus*, and periodically by *Asterionella formosa* and *Cyclotella meneghiniana* as codominants. Throughout the study, the single most successful summer (June-September) species was *Ceratium hirundinella* (Pyrrhophyta). *Ceratium* was the dominant or codominant (i.e., accounted for >10 percent of the biomass) in 88 percent of the summer samples. Overall, it occurred at concentrations >1 g/m³ fresh weight in 89 of the 198 samples collected during the investigation.

36. Summer blue-green algae biomass was usually maintained through a succession of several species, i.e., *Aphanizomenon flos-aquae* -----> *Anabaena planctonica* -----> *Oscillatoria agardhii*. In 1987 and 1988, *A. planctonica* development preceded *A. flos-aquae*, but *O. agardhii* always followed the other

two. *Aphanizomenon flos-aquae* was the most important of the blue-green algae contributing to summer biomass. Autumn diatoms were most obviously represented by *Fragilaria crotonensis*, *A. formosa*, and occasionally by a reoccurrence of *S. hantzschii*.

37. Chlorophyta was the most taxonomically diverse algal group in the plankton, followed in decreasing order by the relatively less diverse groups, Bacillariophyceae and Cyanophyta (Table 4). Despite its high taxonomic diversity (Table 4), Chlorophyta was rarely an important contributor to community biomass (Table 3), and therefore had only a minor effect on the diversity index (H'). In contrast, the abundance of a single species, *C. hirundinella*, within the taxonomically nondiverse group Pyrrophyta (Figure 4) was often so great that it controlled H' (see below).

38. During the 8-year study, summer phytoplankton diversity ranged from 0.05 (1986) to 2.3 (1984). Average summer diversity was greatest during the first 4 years but dropped sharply in 1985 during the extended and massive *Ceratium* bloom (Figure 11). Diversity tended to remain low after 1985, as *Ceratium* continued to account for more than 50 percent of the total phytoplankton biomass. *Ceratium* biomass accounted for somewhat greater variation in diversity than did total biomass. However, *Ceratium* biomass and total biomass were highly correlated throughout the study ($r = 0.96$, $P < 0.001$). Overall, there was a significant negative relationship between biomass and diversity ($r = -0.60$, $P < 0.001$).

39. Evenness (J), a measure of how closely H' approaches the theoretical maximum diversity on a scale of 0 to 1 (Pielou 1966), ranged from 0.18 to 0.73 in summer samples from all years. Mean summer evenness for each study year was always less than 0.5, i.e., 50 percent of maximum diversity (Figure 11), an indication that community diversity was usually well below its potential, further reflecting the importance of only a few species in the community.

PART IV: DISCUSSION

40. Before alum treatment, both elevated epilimnetic TP mass and increases in chlorophyll *a* were common in Eau Galle Reservoir during periods when external TP loading was minimal, indicating that internal sources of TP were responsible for these increases. The occurrence of SRP increases above the profundal sediment surface following the onset of anoxia and the accumulation of SRP in the anoxic hypolimnion indicated that P release from sediments was an important mechanism of internal P loading. Disassociation of iron-phosphorus complexes under anoxic conditions has been suggested as an important mechanism of sediment P release in this reservoir (James, Kennedy, and Gaugush, in press). Investigations conducted in a variety of lakes have shown that P released from anoxic sediments can account for a substantial portion of the internal TP load and TP mass increase in the water column (Theis and McCabe 1978; Larson, Schultz, and Malueg 1981; Nurnberg 1984, 1987; Riley and Prepas 1984; Nurnberg et al. 1986).

41. In lakes where P concentration gradients are present in the metalimnion, P flux via turbulent eddy diffusion often constitutes an important source to the epilimnion (e.g., Imboden and Emerson 1978, Robards and Ward 1978, Stefan and Hanson 1981, Wodka et al. 1983). However, Stauffer (1985) found that this mechanism of P transport was not important in Green Lake, Wisconsin, during late summer because of the absence of P gradients in the metalimnion. The SRP and TP concentration gradients were minimal within the metalimnion (i.e., between approximately the 3- and 6-m depths, Figure 4) of Eau Galle Reservoir, due largely to withdrawal of cooler water during the stratified period. Thus, upward movement of P to the epilimnion via turbulent eddy diffusion was probably minor in this reservoir.

42. Increases in epilimnetic TP mass and chlorophyll *a* have been observed in Eau Galle Reservoir following periods of decreased thermal stability, when wind-driven mixing results in a downward depression of the thermocline (Gaugush 1984). Entrainment of P from deeper water or from metalimnetic shelf sediments is an important mechanism for P transport in Eau Galle Reservoir. Several investigations have reported the occurrence of wind-induced metalimnetic migrations with accompanying increases in epilimnetic P and algal blooms (Stauffer and Lee 1973; Kortmann et al. 1982; Stauffer and Armstrong 1984; Stauffer 1985, 1987; Effler et al. 1986). Biological uptake

and movement of P to the epilimnion by vertically migrating phytoplankton (Salonen, Jones, and Arvola 1984; Osgood 1988; Taylor, Barko, and James 1988) may also be an important transport mechanism. As reported here, the vertically migrating dinoflagellate *Ceratium hirundinella* usually dominates the phytoplankton assemblage in this reservoir during the summer.

43. While alum treatment resulted in a substantial reduction in hypolimnetic TP and internal TP loading in Eau Galle Reservoir during 1986, the abnormally high frequency of major external TP loading events clearly subsidized the mass of epilimnetic TP. Epilimnetic TP mass and chlorophyll *a* remained essentially unchanged from other years. Whereas during most years, particularly before treatment, inflows were quite minor during the stratified period, an exceptionally high year for precipitation and inflow occurred in 1986. Elevated external TP loading clearly mitigated the effects of reduced internal TP loading (due to alum treatment) on epilimnetic TP mass in 1986. Had alum been applied during dry years (e.g., 1982 and 1988), when internal TP loading accounted for most of the TP mass in the reservoir, control over epilimnetic TP mass and therefore chlorophyll *a* (phytoplankton biomass) might have been more substantial than in 1986.

44. The effectiveness of alum treatment in controlling internal TP loading from profundal sediments in Eau Galle Reservoir was greatly diminished by 1987. We suggest that burial of alum by sedimentation associated with high riverine loadings in 1986 was a most important factor in shortening the effectiveness of alum treatment in this system. Eau Galle Reservoir has an average sedimentation rate of 2 to 3 cm/year in the deep basin (James and Barko, in press). The sedimentation rate during 1986 (unmeasured) was probably considerably greater than this average rate. Foy (1985) also suggested that sedimentation was responsible for the rapid recovery of hypolimnetic P concentrations to pretreatment levels following alum treatment in White Lough. In contrast, effective control of sediment P release and TP concentration in the water column of natural lakes with lower sedimentation rates has ranged from 3 to 8 years (Born 1979, Cooke et al. 1982, Garrison and Knauer 1983). As a result, these lakes have experienced reduced levels of algal production and a shift to less noxious species of algae.

45. Recent investigations indicate that littoral sediments may release substantial amounts of P under aerobic conditions, particularly at high pH (Twinch and Peters 1984, Drake and Heaney 1987). In addition, aquatic

macrophytes, prominent in Eau Galle Reservoir (Filbin and Barko 1985), can mobilize sediment P (e.g., Barko and Smart 1980, Carignan 1985, Smith and Adams 1986) that may then be transported to the open water following macrophyte decomposition (e.g., Carpenter 1980, Landers 1982). Prentki et al. (1979) suggested that movement of P from the littoral zone accounted for 59 percent of the soluble P load to the open water of Lake Wingra. Since the shallow-water sediments of the Eau Galle Reservoir were not treated in this study, it is possible that epilimnetic TP mass during low external loading periods was influenced by internal TP loading from these areas.

46. Ungauged sources of TP, particularly from ground-water discharge, may provide another internal source of P to the epilimnion of Eau Galle Reservoir. Indeed, hydraulic discharge from this reservoir has exceeded water income throughout our studies. This imbalance was particularly apparent during the relatively dry summers of 1982 and 1988 (Figure 2). Cooke et al. (1977, 1982) suggested that ground-water flow through P-rich sediments may be an important internal P-loading mechanism in West Twin and Dollar Lakes in Ohio.

47. Although internal P loading predominates during the summer stratified period in Eau Galle Reservoir, episodic freshets frequently result in substantial external P inputs to the epilimnion. Both P sources are important in sustaining high concentrations of TP and chlorophyll *a* in this reservoir. Conversely, internal P loading appears to be much more important than external P loading in natural lakes during the summer stratified period (e.g., Cooke et al. 1977; Larson, Schultz, and Malueg 1981; Stauffer 1987). In comparison with natural lakes, reservoirs receive water and material inputs from tributaries that drain a relatively larger watershed (Baxter 1977, Thornton et al. 1981). This difference results in generally greater nutrient and sediment loads to reservoirs, making them more difficult to manage than natural lakes via P control measures.

48. As with many reservoirs, Eau Galle Reservoir has a small surface area and volume relative to the size of its drainage basin. Consequently, flushing of the pool occurs regularly with the melting of snow and ice in the spring, and with storm events throughout the year. High periodic flushing rates combined with agricultural/dairy land-use practices and local geology ensure excessive nutrient loading to the system. As a consequence of these

factors, Eau Galle Reservoir remains eutrophic and supports a large phytoplankton population composed of nuisance species.

49. The seasonal succession of phytoplankton in Eau Galle Reservoir is fundamentally similar to that described for many temperate eutrophic lakes in which spring diatoms give way to blue-green or dinoflagellate summer populations, which in turn are replaced by diatoms in the autumn (Hutchinson 1967, Round 1981). The summer phytoplankton community structure in Eau Galle Reservoir is a eutrophic Pyrrophyta/Cyanophyta association, a combination of Hutchinson's (1967) phytoplankton association categories.

50. No significant difference in phytoplankton biomass between pre-treatment and posttreatment years was noted except in 1985. The high phytoplankton biomass in the summer of 1985 (90 g/m^3 wet weight) is remarkable from several points of view. Rarely have such high values been reported for any lake, and in no reported case has *Ceratium hirundinella* accounted for biomass of this magnitude (Nicholls, Kennedy, and Hammett 1980). Notably, the 1985 (*Ceratium*) bloom followed a year in which the late-summer phytoplankton community was totally dominated by blue-green algae.

51. The explanation for the abrupt change in dominance from blue-green algae to *Ceratium* in 1985 (persisting to date) may lie partially in the differences in the life histories of the dominant species involved, relative to the effects of winter drawdown in 1984. Akinetes (*Aphanizomenon*) and cysts (*Ceratium*) resting on the sediment surface provide a seasonal inoculum for these species (Wildman, Loescher, and Winger 1975; Heaney, Chapman, and Morison 1983; respectively). While the germination of cysts appears to require only oxygen and temperatures a greater than about 4°C (Heaney, Chapman, and Morison 1983; Loeblich and Loeblich 1984; Anderson, Taylor and Armbrust 1987), warmer water temperatures and a light cue, in addition to oxygen, are required for akinete germination (Wildman, Loescher, and Winger 1975). Thus, *Ceratium* has the potential to derive from sediments at any depth during periods of mixing. In contrast, due primarily to its requirement for light in akinete germination, *Aphanizomenon* is most likely to derive from sediments of the littoral zone during periods of mixing in Eau Galle Reservoir. As a result, the winter drawdown, which impacted sediments in the littoral zone but not in deeper water, probably had a much more damaging effect on *Aphanizomenon* than on *Ceratium*. In addition, since buoyant vegetative filaments of *Aphanizomenon* can persist well into the winter (Jones 1979; Wildman, Loescher, and Winger

1975), it is possible that direct loss of cells of this species accompanied loss (about 47 percent) of pool volume during winter drawdown.

52. The 1985 *Ceratium* bloom may have also been indirectly related to the winter drawdown. The drawdown had a direct effect on the submersed macrophyte community, which may in turn have affected the phytoplankton. *Ceratophyllum demersum*, which earlier accounted for 58 percent of the total macrophyte standing crop (Filbin and Barko 1985), was severely diminished in its distribution, standing crop, and frequency of occurrence (Godshalk and Barko 1988). This species has been shown to be particularly sensitive to cold exposure, and its growth is often at least temporarily retarded following reduction in water levels (e.g., Beard 1973, Nichols 1975). *Ceratophyllum* (without a true root system) and attached epiphytes derive their nutrient supply from the water column, and are therefore in direct competition for nutrients with phytoplankton.

53. In the years prior to the drawdown, the large standing stock of macrophytes and epiphytes formed a reservoir of P and other nutrients unavailable to the phytoplankton during much of the stratified period (Filbin and Barko 1985). In 1985, this nutrient sink was much reduced, perhaps providing greater access to nutrient supplies by the phytoplankton community.

54. As discussed above, abundant P is available to the phytoplankton community of Eau Galle Reservoir from multiple sources. Alum treatment clearly and significantly reduced P supply from profundal sediments in 1986 but had no effect (except perhaps relative to 1985) on phytoplankton production. Unfortunately, the greatest number of summer storm events during the entire study was recorded in 1986. Precipitation during June and July was 2 to 3 times greater than during these months in early years. As a result, high external TP loading during the stratified period of 1986 allowed the maintenance of high TP mass and chlorophyll in the epilimnion. Weather-related events in 1986 apparently counteracted the effects of alum treatment.

55. One of the unique features of reservoirs compared with natural lakes is their physical and chemical instability (Baxter 1977). Eau Galle Reservoir is particularly unstable in these regards, due to its relatively large watershed and its small pool volume. Superimposed upon this inherent instability were major extremes in climate (wet versus dry years) and a winter drawdown (1984) during the 8-year study period. In spite of these extremes, plus the additional effects of alum treatment (1986) on chemical

conditions/gradients, the phytoplankton community has been remarkably stable and of low species diversity. A shift in biomass toward a single species (*Ceratium*) after 1984 further decreased species diversity. *Ceratium* appears to be particularly well suited to existence in an unstable environment. The switch in relative dominance from a mixed dinoflagellate/blue-green algae community to one dominated by *Ceratium* is notable. However, through 1988 the status of these phytoplankton groups as codominants did not change. Without a decrease in P availability, *Ceratium* and blue-green algae are likely to persist as codominants in Eau Galle Reservoir, but perhaps will vary in relative abundance from year to year.

PART V: FUTURE STUDIES

56. Since internal loadings of P in Eau Galle Reservoir probably cannot be controlled over the long term by alum application alone, other means of reducing the availability of P to phytoplankton need to be developed. For Eau Galle Reservoir and other systems with selective withdrawal features, the opportunity to release water from specific depths provides a potentially powerful tool for reconfiguring thermal structure. Particularly in shallow systems such as Eau Galle reservoir, thermal structure has an important influence on the movement of nutrients from bottom waters to the epilimnion.

57. At present, with hypolimnetic withdrawal, weak vertical gradients in water temperature lend Eau Galle reservoir to wind-driven mixing. As discussed earlier, episodic mixing events give rise to phytoplankton blooms by intermixing P-rich bottom waters with the epilimnion. We are currently investigating the effects of a change in withdrawal from bottom to surface during the summer. The rationale for this change in reservoir operation is that nutrient recycling associated with weak thermal structure and hypolimnetic warming (Gaugush 1984; James, Kennedy, and Gaugush, in press) has contributed to past phytoplankton blooms in Eau Galle reservoir (Barko et al. 1984).

58. Effects on physical conditions following change in withdrawal (scheduled for summer 1990) are anticipated to include decreased temperature of the hypolimnion, more stable thermal stratification, greater water column stability, lesser intermixing of bottom waters with the epilimnion, and presumably, reduced phytoplankton productivity. Ultimate effects on phytoplankton productivity will depend on species composition, which may also be altered by change in withdrawal.

59. During 1991 and 1992, a combined approach will be implemented in an attempt to manage phytoplankton population densities and species composition in Eau Galle reservoir. This approach will derive from results obtained thus far on independent effects of midwinter drawdown and alum treatment, and in addition, independent effects (to be determined) of change in reservoir operation from bottom to surface withdrawal. Concurrent with ongoing studies of phytoplankton dynamics, investigations of phosphorus cycling, with attention to various internal and external sources of input to the reservoir, will continue.

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Table 1
Mean (± 1 Standard Error) Daily Internal and External TP Loadings
to Eau Galle Reservoir During Midsummer Periods (June-August)

Year	Internal TP Loading	n	ANOVA*	External TP Loading	n	ANOVA
	$\text{mg m}^{-2} \text{ day}^{-1}$			$\text{mg m}^{-2} \text{ day}^{-1}$		
1981	9.31 (4.13)	7	a	2.76 (1.14)	8	b
1982	7.84 (2.87)	8	a	1.28 (0.01)	92	b
1986	-5.58 (6.21)	6	b	17.01 (5.67)	85	a
1987	6.25 (2.60)	9	ab	4.89 (1.96)	65	b
1988	8.97 (5.01)	8	a	1.29 (0.03)	85	b

Note: Alum was applied to the reservoir in 1986.

* Analysis of variance column summarizes results of Duncan's Multiple Range Test. Different letters associated with individual variables indicate significant differences at the 5-percent level or less.

Table 2
Mean (± 1 Standard Error) Chlorophyll *a* and TP Mass in Epilimnion
(0 to 3 m) of Eau Galle Reservoir, June-August

Year	Chlorophyll <i>a</i>	n	ANOVA*	TP Mass	n	ANOVA
	mg m^{-3}			kg		
1981	35.99 (6.51)	7	a	126.70 (14.06)	7	a
1982	50.15 (10.90)	6	a	108.05 (13.48)	8	a
1986	28.79 (7.88)	7	a	148.27 (18.98)	8	a
1987	56.38 (12.22)	11	a	163.13 (22.00)	12	a
1988	40.22 (7.38)	8	a	110.28 (18.72)	9	a

Note: Alum was applied to the reservoir in 1986.

* Analysis of variance column summarizes the results of Duncan's Multiple Range Test. Different letters associated with individual variables indicate significant differences at the 5-percent level or less.

Table 3

Biomass Attained by Dominant Phytoplankton Species in
Eau Galle Reservoir, 1981-1988

Species	Group*	Biomass Classes				Total Occurrences >1 g m ⁻³ , fresh weight	Season
		>20	g m ⁻³ , 10-20	5-10	1-5		
<i>Ceratium hirundinella</i>	pyr	21	20	19	2	89	Summer
<i>Stephanodiscus hantzschii</i>	bac	6	6	3	10	25	Spring, autumn
<i>Aphanizomenon flos-aquae</i>	cya	--	3	3	34	40	Summer
<i>Oscillatoria agardhii</i>	cya	--	1	1	11	13	Summer
<i>Asterionella formosa</i>	bac	--	1	--	7	8	Spring, autumn
<i>Anabaena planctonica</i>	cya	--	--	12	16	28	Summer
<i>Fragilaria crotonensis</i>	bac	--	--	1	17	18	Summer, autumn
<i>Cryptomonas erosa</i>	cry	--	--	1	15	16	All seasons
<i>Rhodomonas minuta</i>	cry	--	--	--	6	6	Spring, summer, autumn
<i>Cyclotella meneghiniana</i>	bac	--	--	--	5	5	Spring
<i>Stephanodiscus inviscitatus</i>	bac	--	--	--	4	4	Spring
<i>Anabaena spiroides</i>	cya	--	--	--	3	3	Summer
<i>Anabaena flos-aquae</i>	cya	--	--	--	2	2	Summer
<i>Cryptomonas marssonii</i>	cry	--	--	--	2	2	Spring, autumn
<i>Melosira granulata</i>	bac	--	--	--	2	2	Autumn
var. <i>angustissima</i>	bac	--	--	--	1	1	Summer
<i>Rhizosolenia eriensis</i>	bac	--	--	--	1	1	Autumn
<i>Cyclotella pseudostelligera</i>	bac	--	--	--	1	1	Spring
<i>Chrysochromulina parva</i>	chr	--	--	--	1	1	Spring
<i>Pandorina morum</i>	chl	--	--	--	1	1	Summer
<i>Sphaerocystis Schroeteri</i>	chl	--	--	--	1	1	Summer

* pyr = Pyrrophyta, bac = Bacillariophyceae, cya = Cyanophyta, cry = Cryptophyta, chr = Chrysophyta, chl = Chlorophyta.

Table 4
Number of Phytoplankton Species, by Major Group, Identified
in Eau Galle Reservoir, Wisconsin

<u>Taxon</u>	<u>Number of Species</u>	<u>Number of Dominant Species*</u>
Chlorophyta	53	2
Bacillariophyceae	34	8
Cyanophyta	27	5
Chrysophyta	9	1
Cryptophyta	7	4
Pyrrhophyta	3	1
Euglenophyta	<u>3</u>	<u>0</u>
	136	21

* A species was considered dominant if it occurred in at least one sample at a biomass $>1 \text{ g m}^{-3}$ (see Table 3).

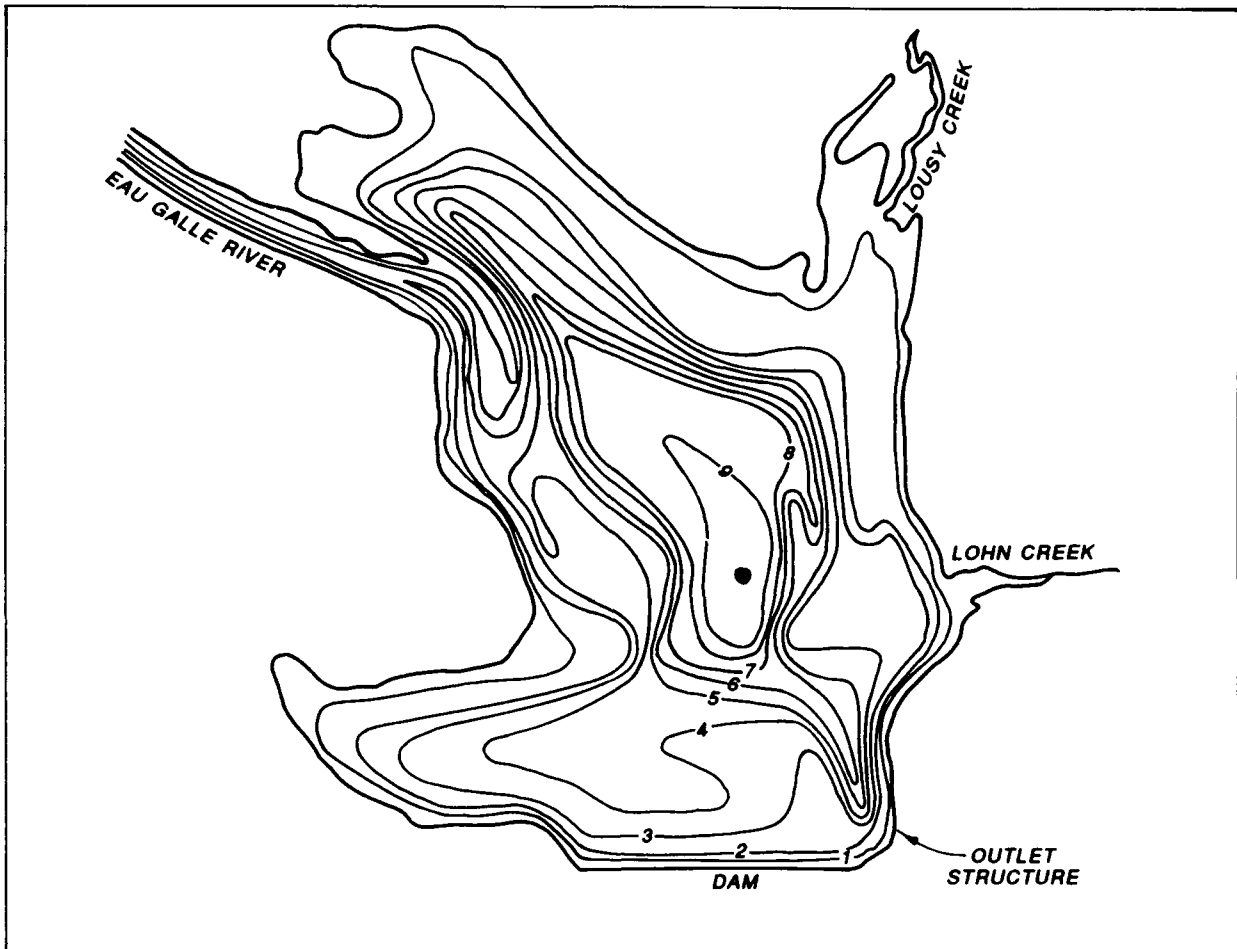


Figure 1. Morphometric map of Eau Galle Reservoir. Contours indicate depth (in meters). Solid circle represents sampling site in the reservoir. French Creek, located upstream on the Eau Galle River, is not shown

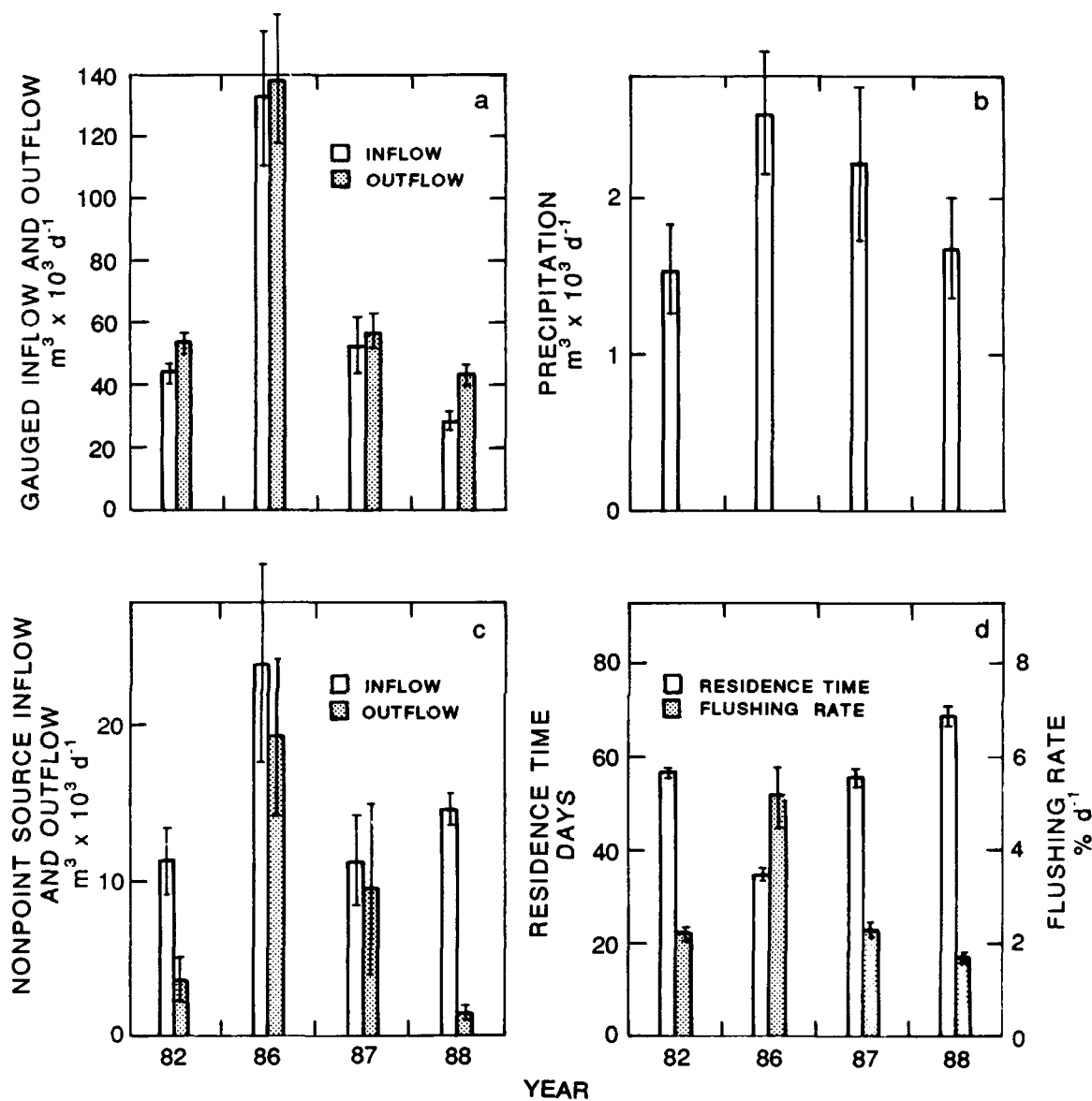


Figure 2. Hydrologic data (± 1 standard error, $n = 153$) for Eau Galle Reservoir, May through September: (a) mean daily gauged inflow (includes Eau Galle River, Lousy, Lohn, and French Creeks, except in 1988 when the creeks were not gauged) and outflow, (b) mean daily precipitation, (c) mean daily ungauged (nonpoint) inflows and outflows, and (d) mean daily residence time and flushing rate

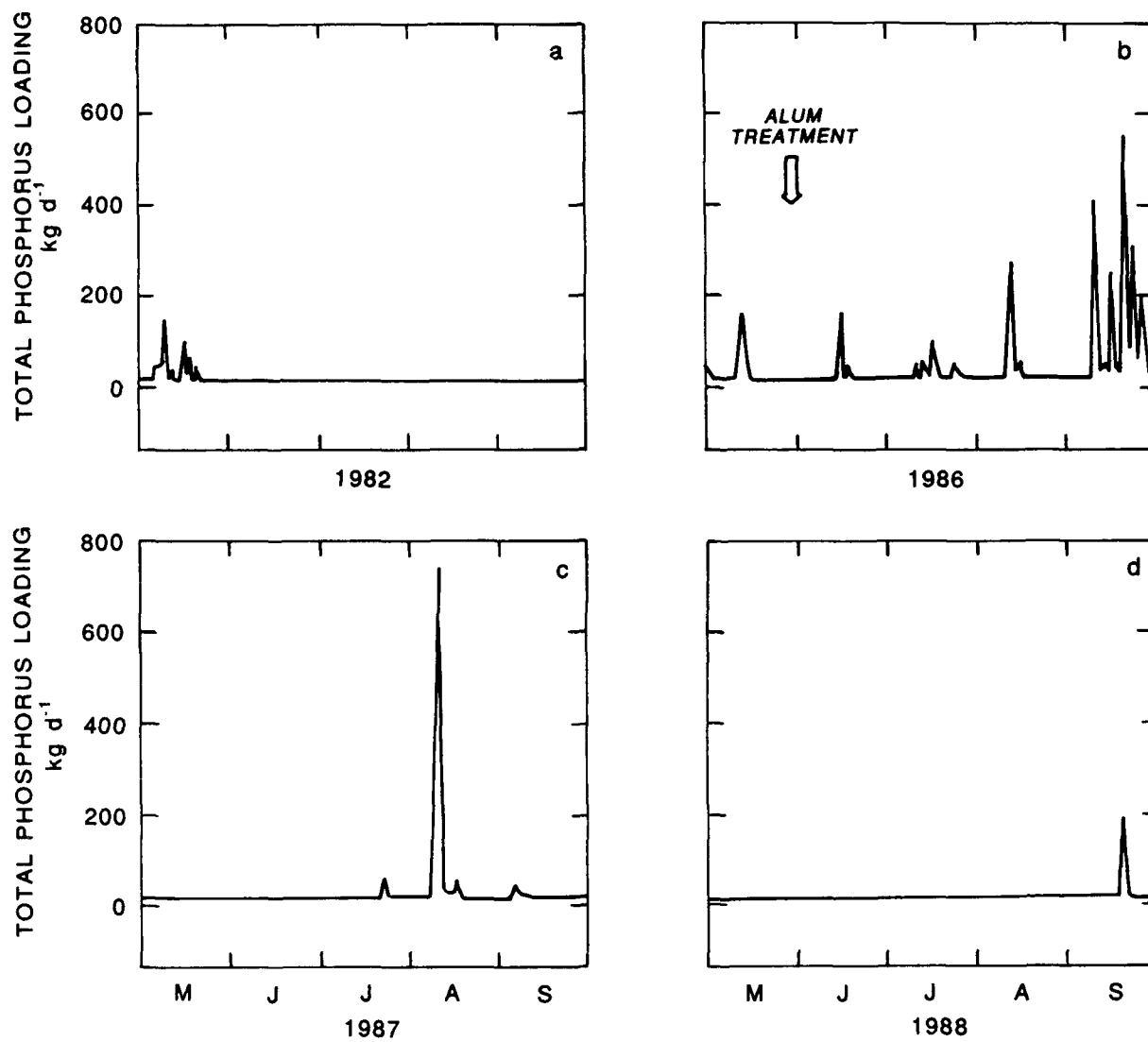


Figure 3. Daily TP loading from the Eau Galle River, May-September 1982, 1986, 1987, and 1988

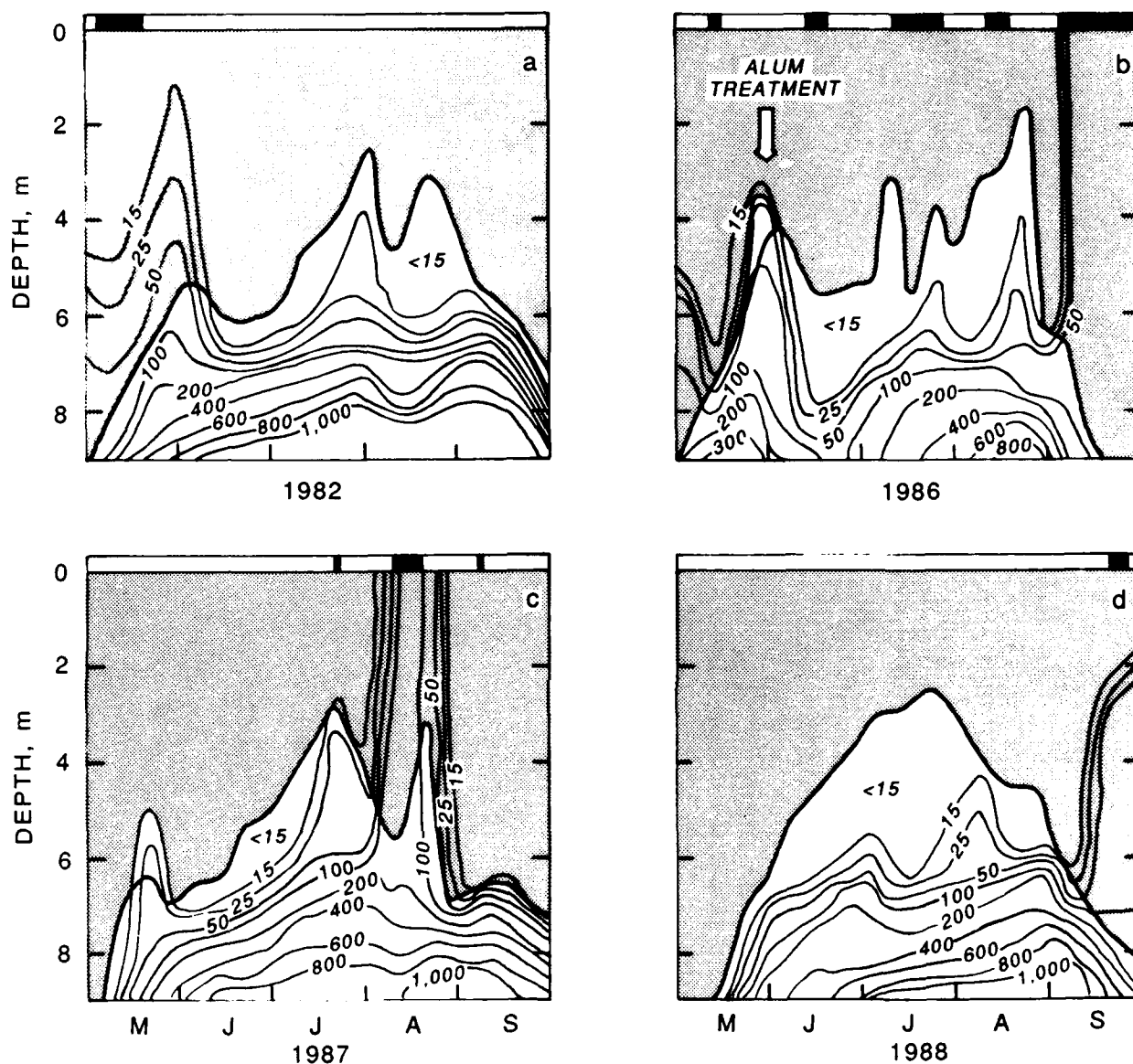


Figure 4. Seasonal and depth-related variations in soluble reactive phosphorus ($\mu\text{g}/\text{l}$). Measurements were made at close depth intervals (25 cm) during May-September 1982, 1986, 1987, and 1988. Shaded areas indicate zones in which dissolved oxygen was >0.5 mg/l. Black areas above panels indicate periods during which external TP loading exceeded 25 kg TP/day

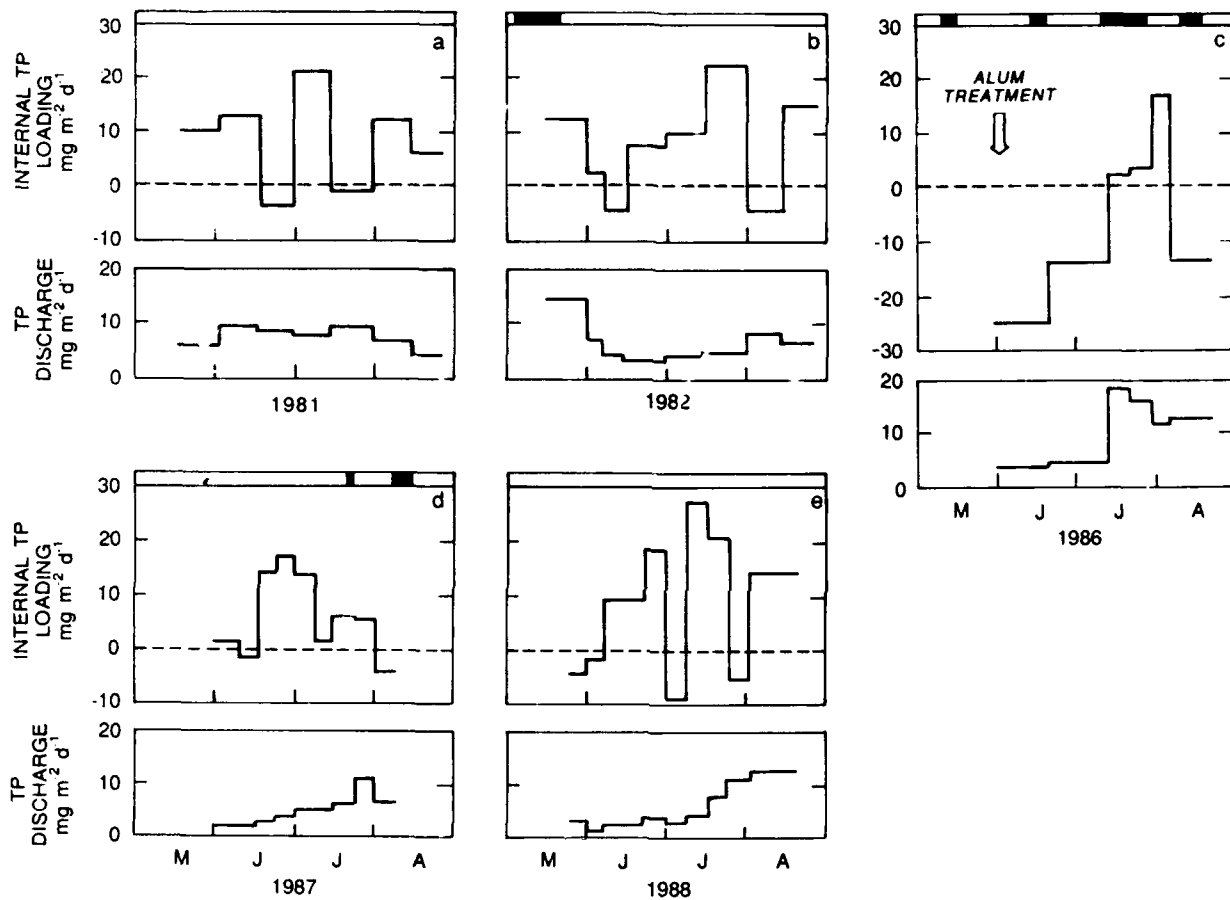


Figure 5. Seasonal variations in internal TP loading (upper panels) and TP discharge (lower panels) for the years 1981, 1982, 1986, 1987, and 1988. Dashed lines indicate an internal TP loading rate of zero. Black areas above panels indicate periods during which external TP loading exceeded 25 kg TP/day

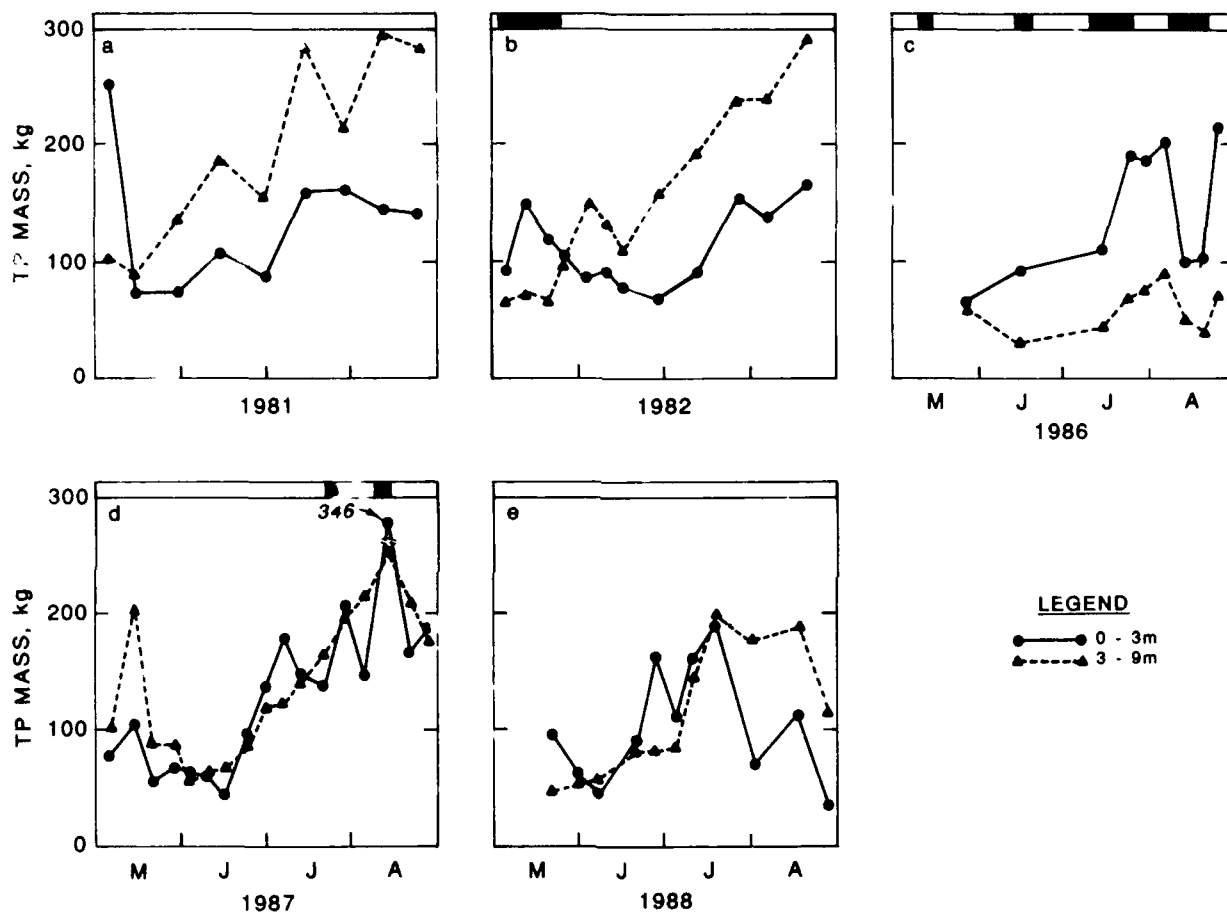


Figure 6. Seasonal variations in TP mass within the epilimnion (0 to 3 m) and bottom waters (3 to 9 m) during 1981, 1982, 1986, 1987, and 1988. Black areas above each panel indicate periods during which external TP loading exceeded 25 kg TP/day

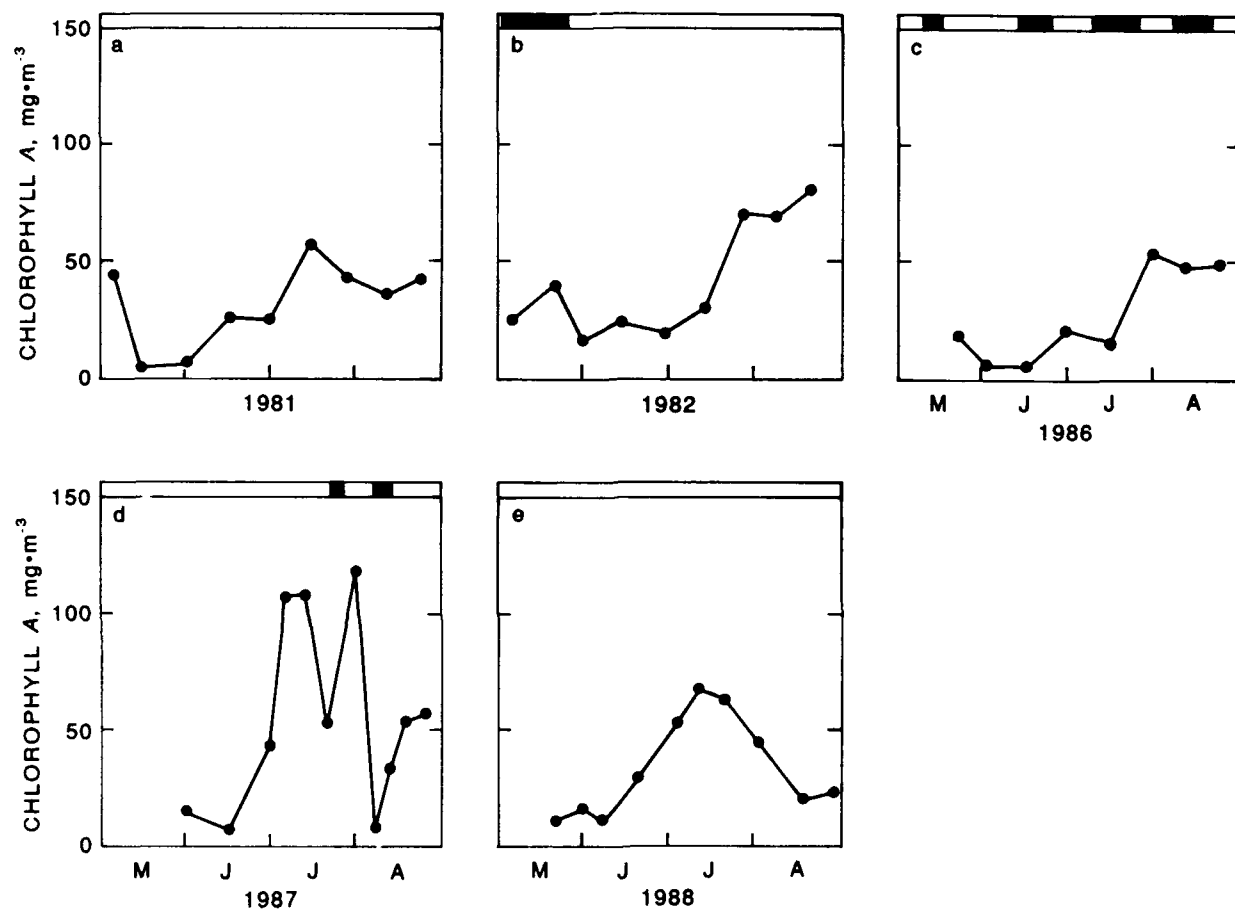


Figure 7. Seasonal variations in epilimnetic (0 to 3 m) chlorophyll *a* concentrations during 1981, 1982, 1986, 1987, and 1988. Black areas above each panel indicate periods during which external TP loading exceeded 25 kg TP/day

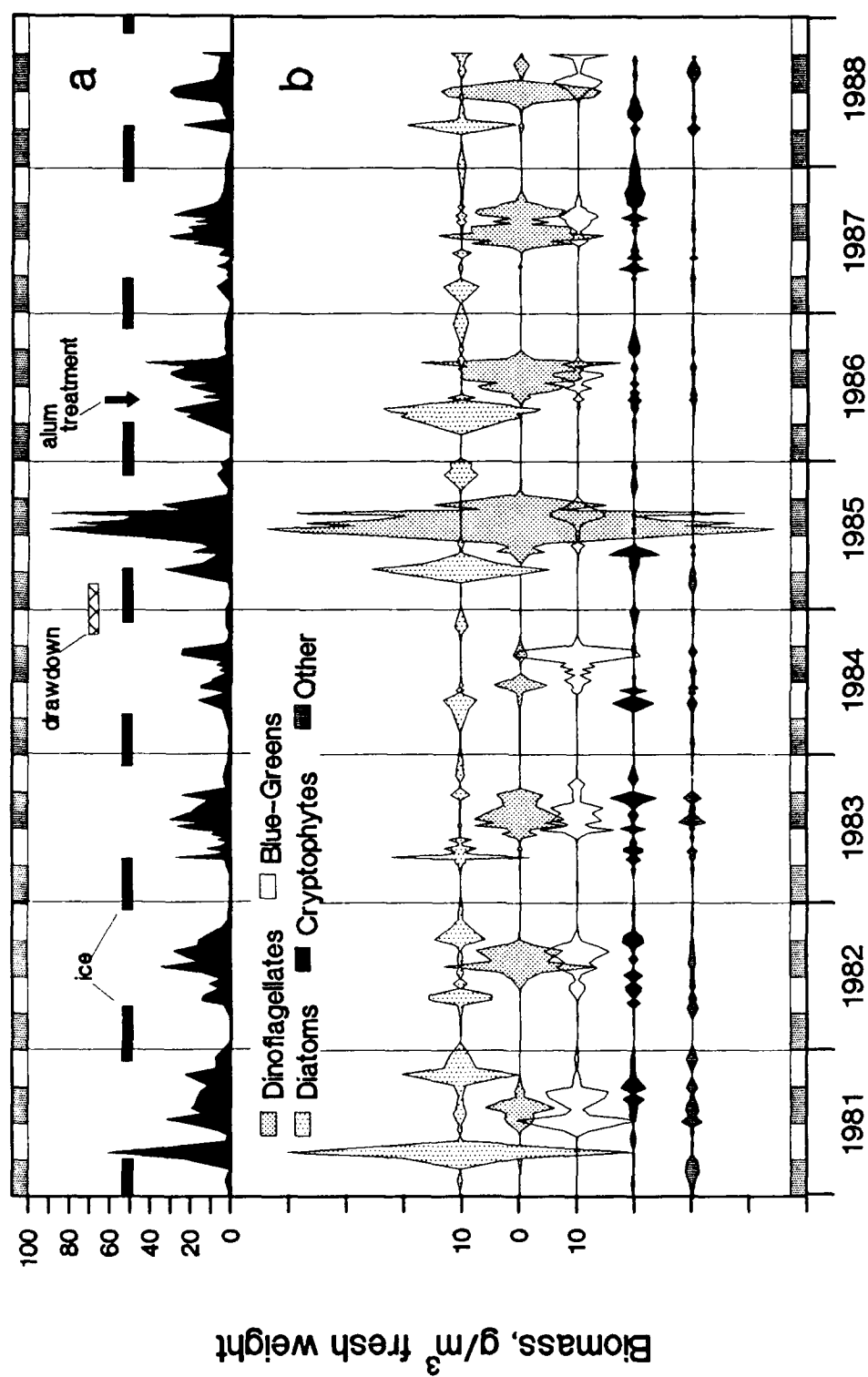


Figure 8. Total seasonal phytoplankton biomass (a) and biomass by major groups (b) for the years 1981 through 1988

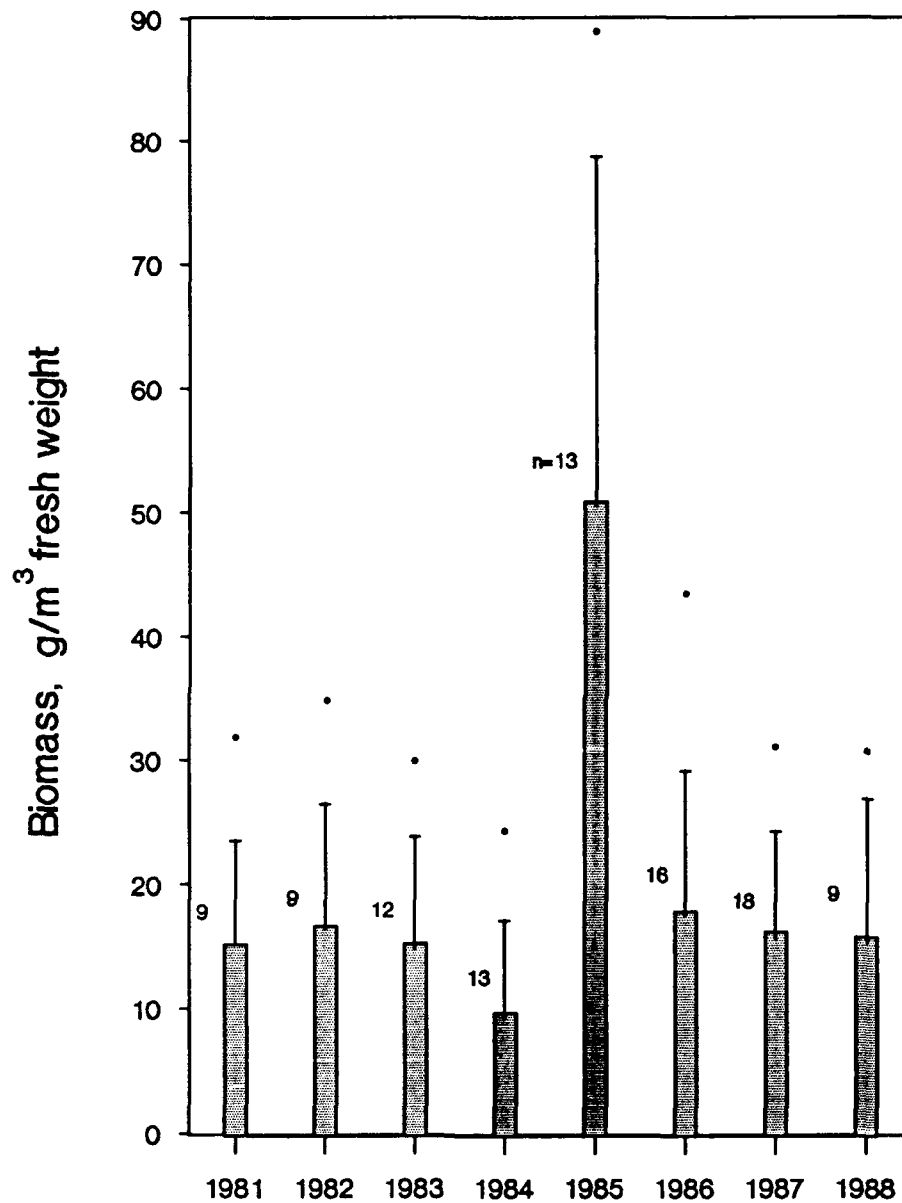


Figure 9. Mean summer (June-September) phytoplankton biomass (g m^{-3} fresh weight), with notations of 1 standard deviation, n , and the maximum summer biomass (denoted by filled circles)

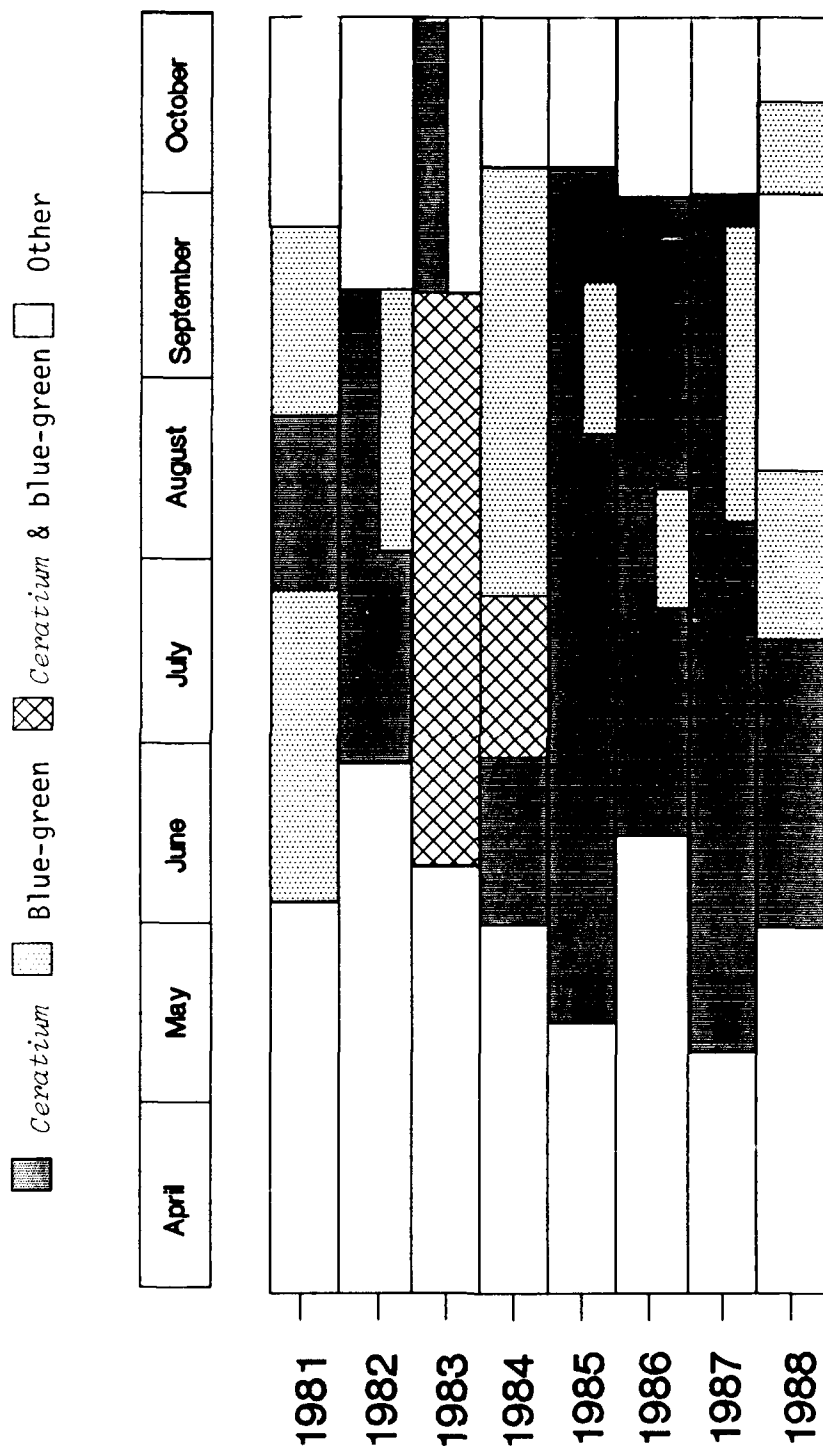


Figure 10. Summary of between-year differences in seasonal succession of *Ceratium* and blue-green algae

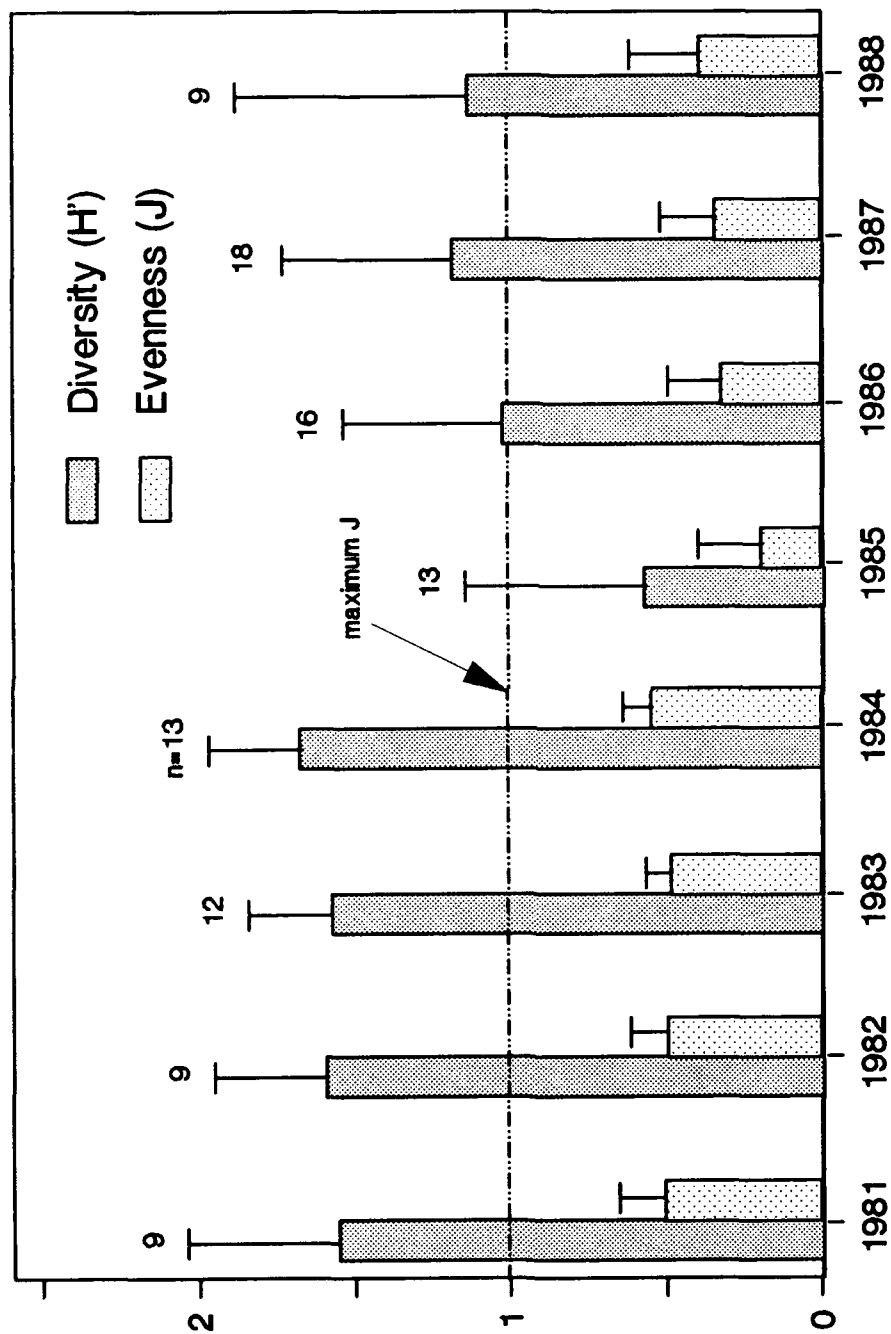


Figure 11. Mean summer (June-September) phytoplankton diversity (H') and evenness (J) with 1 standard deviation and n

APPENDIX A
PHYTOPLANKTON SPECIES LIST, EAU GALLE RESERVOIR
(1981-1988)

BACILLARIOPHYCEAE

Achnanthes
 minutissima Kuetzing
Amphora sp.
Asterionella
 formosa Hassall
Cocconeis
 placentula
 v. *lineata* (Ehr.) Van Heurck
Cyclotella
 meneghiniana Kuetzing
 pseudostelligera Hustedt
 stelligera Cleve & Grunow
Cymatopleura
 solea (Bréb.) W. Smith
Cymbella sp.
Fragilaria
 capucina Desmazieres
 capucina
 v. *mesolepta* Rabenhorst
 crotonensis Kitton
 vaucheriae (Kuetz.) Petersen
Gomphonema sp.
Gyrosigma
 spencerii (W. Smith) Cleve
Melosira
 crenulata (Ehr.) Kuetzing
 granulata (Ehr.) Ralfs
 granulata
 v. *angustissima* Mueller
 varians C. A. Agardh
Navicula
 minima Grunow
Nitzschia
 acicularis W. Smith
 actinastroides (Lemm.) Van Goor
 palea (Kuetz.) W. Smith
 sigmoidea (Ehr.) W. Smith
 vermicularis (Kuetz.) Grunow
Rhizosolenia
 eriensis H. L. Smith
Stephanodiscus
 hantzschii Grunow
 invisitatus Hohn & Hellerman

Synedra

acus Kuetzing
 delicatissima
 v. *angustissima* Grunow
 rumpens Kuetzing
 ulna (Nitz.) Ehrenberg
 ulna
 v. *danica* (Kuetz.) Van Heurck
 ulna
 v. *subaequalis* (Grun.) Van Heurck

CHLOROPHYTA

Actinastrum
 hantzschii Lagerheim
 hantzschii
 v. *fluviatile* Schroeder
Ankistrodesmus
 convolutus Corda
 falcatus (Corda) Ralfs
 falcatus
 v. *mirabilis* (W. & W.) G. S. West
 gelifactum (Chod.) Bourrelly
 nannoselene Skuja
Carteria sp.
Chlamydomonas
 snowii Printz
Chlorogonium sp.
Closteriopsis sp.
Closteridium sp.
Closterium sp.
Coelastrum
 cambricum Archer
 microporum Naegeli
 sphaericum Naegeli
Cosmarium sp.
Crucigenia
 truncata G. M. Smith
Dictyosphaerium
 ehrenbergianum Naegeli
 pulchellum Wood
Elakatothrix
 gelatinosa Wille

APPENDIX A (Continued)

Eudorina
 elegans Ehrenberg
Gloeocystis
 planctonica (W. & W.) Lemmermann
Kirchneriella sp.
Micractinium
 pusillum Fresenius
 pusillum
 v. *elegans* G. M. Smith
Mougeotia sp.
Oedogonium sp.
Oocystis sp.
Pandorina
 morum (Muell.) Bory
Pediastrum
 boryanum (Turp.) Meneghini
 duplex
 v. *clathratum* (A. Braun) Lagerheim
 duplex
 v. *reticulatum* Lagerheim
Quadrigula
 chodatti (Tanner-Fullman) B. M. Smith
 lacustris (Chod.) G. M. Smith
Scenedesmus
 abundans (Kirch) Chodat
 acuminatus (Lag.) Chodat
 arcuatus
 v. *platydisca* G. M. Smith
 bijuga (Turp.) Lagerheim
 dimorphus (Turp.) Kuetzing
 dispar Brébisson
 obliquus (Turp.) Kuetzing
 quadricauda (Turp.) Brébisson
 quadricauda
 v. *longispina* (Chod.) G. M. Smith
 quadricauda
 v. *parvus* G. M. Smith
Schroederia
 judayi G. M. Smith
 setigera (Schroed.) Lemmermann
Selenastrum
 minutum (Naeg.) Collins
Sphaerocystis
 schroeteri Chodat
Stauroastrum
 paradoxum Meyen
Tetraëdron sp.
Tetrastrum
 staurogeniaeforme (Schroed.) Lemmermann

Treubaria
 setigerum (Archer) G. M. Smith

CHRYSTOPHYTA

Chrysochromulina
 parva lackey
Chrysococcus
 refuscens klebs
Dinobryon sp.
Kephyrion sp.
Mallomonas sp.
Ochromonas sp.
Syncrypta
 volvox Ehrenberg
Synura
 uvella Ehrenberg
Stenokalyx
 inconstans G. Schmid

CRYPTOPHYTA

Chroomonas sp.
Cryptomonas
 erosa Ehrenberg
 marssonii Skuja
 ovata Ehrenberg
Ketoblepharis
 ovalis Skuja
Rhodomonas
 minuta Skuja
 minuta
 v. *nannoplanctica* Skuja

CYANOPHYTA

Anabaena
 affinis Lemmermann
 flos-aquae (Lyngb.) Brébisson
 planctonica Brunnthaler
 spiroides Klebahn
 spiroides
 v. *crassa* Lemmermann
Anabaenopsis
 raciborskii Woloszyńska

APPENDIX A (Concluded)

CYANOPHYTA (Continued)

Aphanocapsa
delicatissima West & West
pulchra (Kuetz.) Rabenhorst
Aphanizomenon
flos-aquae (L.) Ralfs
Aphanothece
nidulans P. Richter
Chroococcus
minimus (Keissl.) Lemmermann
Coelosphaerium
dubium Grunow
naegelianum Unger
pallidum Lemmermann
Dactylococcopsis
raphidioides Hansgirg
Gomphosphaeria
lacustris
v. compacta Lemmermann
aponina
v. delicatula Virieux
Lyngbya sp.
Merismopedia
minima Beck
tenuissima Lemmermann
Microcystis
aeruginosa Kuetz.; emend. Elenkin
incerta Lemmermann
Oscillatoria
angustissima West & West
agardhii Gomont
limnetica Lemmermann
Phormidium
mucicola Naumann & Huber-Pestalozzi
Spirulina sp.

EUGLENOPHYTA

Euglena sp.
Phacus
pyrum (Ehr.) Stein
Trachelomonas sp.

PYRRHOPHYTA

Ceratium
hirundinella (O.F. Muell.) Schrank
Glenodinium sp.
Gymnodinium sp.